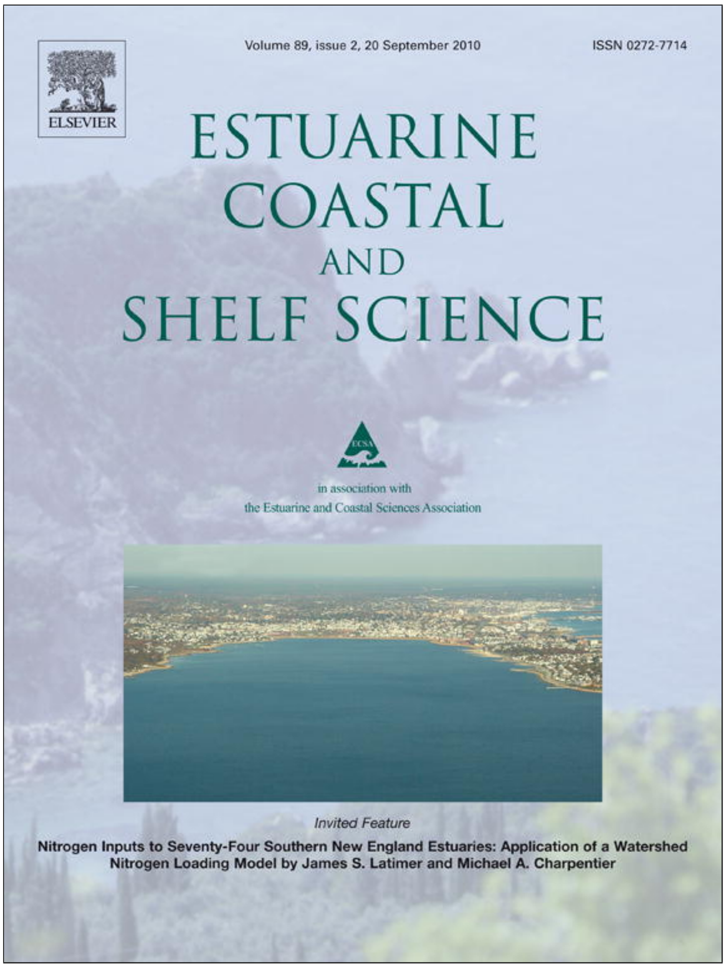
Volume 89, issue 2, 20 September 2010
ISSN 0272-7714
ELSEVIER
ESTUARINE
COASTAL
AND
SHELF SCIENCE
in association with
the Estuarine and Coastal Sciences Association
Invited Feature
Nitrogen Inputs to Seventy-Four Southern New England Estuaries: Application of a Watershed Nitrogen Loading Model by James S. Latimer and Michael A. Charpentier

Estuarine, Coastal and Shelf Science 89 (2010) 125–136

Contents lists available at ScienceDirect

Estuarine, Coastal and Shelf Science

journal homepage: www.elsevier.com/locate/ecss

Invited Feature

Nitrogen inputs to seventy-four southern New England estuaries: Application of a watershed nitrogen loading model

James S. Latimer [a,*], Michael A. Charpentier [b,1]

[a] US Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, Narragansett RI 02882, USA

[b] Raytheon Information Solutions, contractor to the US Environmental Protection Agency, Narragansett RI 02882, USA

ARTICLE INFO

Article history:
Received 29 December 2009
Accepted 7 June 2010
Available online 30 June 2010

Keywords:
estuaries
nitrogen
watersheds
eutrophication
new england

ABSTRACT

Excess nitrogen inputs to estuaries have been linked to deteriorating water quality and habitat conditions which in turn have direct and indirect impacts on aquatic organisms. This paper describes the application of a previously verified watershed loading model to estimate total nitrogen loading rates and relative source contributions to 74 small-medium sized embayment-type estuaries in southern New England. The study estuaries exhibited a gradient in nitrogen inputs of a factor of over 7000. On an areal basis, the range represented a gradient of approximately a factor of 140. Therefore, all other factors being equal, the study design is sufficient to evaluate ecological effects conceptually tied to excess nitrogen along a nitrogen gradient. In addition to providing total loading inputs rates to the study estuaries, the model provides an estimate of the relative contribution of the nitrogen sources from each watershed to each associated estuary. Cumulative results of this analysis reveal the following source ranking (means): direct atmospheric deposition (37%), ≈wastewater (36%), >indirect atmospheric deposition (16%) > fertilizer (12%). However, for any particular estuary the relative magnitudes of these source types vary dramatically. Together with scientific evidence on symptoms of eutrophication, the results of this paper can be used to develop empirical pressure-state models to determine critical nitrogen loading limits for the protection of estuarine water quality.

Published by Elsevier Ltd.

1. Introduction

Nitrogen is an important macronutrient on which the global food supply is dependent. It is also essential to the health and ecological integrity of estuaries. In excessive amounts, however, nitrogen can cause cultural eutrophication, a man-made increase in the rate of supply of organic matter to marine aquatic ecosystems (Nixon, 1995). Formation of reactive nitrogen, that portion that can be used by biological systems, continues to increase every year (Galloway et al., 2008). Excess amounts in estuaries can lead to low dissolved oxygen, fish kills, overabundance of nuisance and harmful algae and macrophytes, loss of vascular plants (i.e., seagrasses), increased sedimentation, and detrimental shifts of both floral and faunal species and other food web modifications (Cloern, 2001).

The need to estimate nitrogen loading to estuaries is therefore acute. More and more municipalities are turning to loading reductions as a means to reduce the adverse effects of cultural eutrophication in the coastal marine environment (CBP, 2000; NYSDEC and CTDEP, 2000; Greening and Janicki, 2006). In cases where there is a need to estimate nonpoint source nitrogen loading over broad regions and many estuaries, site-specific loading methods may not be appropriate due to excessive cost and effort. In instances where the loading is not dominated by monitored riverine and point source inputs, the development of loading estimates may require years of sampling and analysis. In contrast, for these types of estuaries, simple watershed models that allow estimates for many estuaries with more limited data may be the more appropriate approach.

A survey of the literature reveals that there are a number of approaches that have been used to estimate nitrogen loading from watersheds. Most of these approaches suffer from one deficiency or another with respect to the purpose of this study: to obtain simple, first-order estimates of nitrogen loading rates for a large number of small estuaries. For example, the US Geological Survey (USGS) Spatially Referenced Regressions on Watershed Attributes Model (SPARROW) is aimed at estimating nutrient fluxes to stream reaches and can apply only to estuaries that have surface water inputs

* Corresponding author. Tel.: + 1401 782 3167.
E-mail addresses: latimer.jim@epa.gov (J.S. Latimer), charpentier.mike@epa.gov (M.A. Charpentier).
[1] Tel.: +1 401 782 3186.

0272-7714/$ – see front matter Published by Elsevier Ltd.
doi:10.1016/j.ecss.2010.06.006

(Moore et al., 2004). Another approach, called "the Simple Method" (www.stormwatercenter.net), is aimed at estimating pollutants from stormwater runoff and requires, among other things, data on nitrogen concentrations in stormwater from differing land uses. Nitrogen export coefficients have been published for watersheds surrounding estuaries (Reckhow et al., 1980; Frink, 1991); however, the coefficients exhibit extreme spatial variability. Finally, the ArcView based Generalized Watershed Loading Functions is a combined distributed/lumped parameter watershed model that provides continuous simulation with daily time steps for weather and mass balance (Farley and Rangarajan, 2006; Georgas et al., 2009). It requires a considerable amount of site-specific data on weather, hydrology, soil erosion, and surface nitrogen concentration from streams.

The purpose of this paper is to illustrate the application of a previously published and verified nitrogen loading model (NLM) to 74 watershed-estuary systems and to assess whether there were differences in nitrogen loading and relative source strengths for two US ecoregions, namely the Northeast Coastal Zone (NCZ) and the Atlantic Coast Pine Barrens (APB) in southern New England and among the three New England States: Connecticut (CT), Rhode Island (RI) and Massachusetts (MA). The specific watershed-estuary systems were chosen as part of a study to determine the nature of eutrophication responses along a gradient in nitrogen inputs. Thus, by comparing ecological responses for a large number of estuaries, which have many physical attributes in common, but vary according to the magnitude of nitrogen inputs, one can test the hypothesis that the environmental pressure exerted by nitrogen loading is associated with the ecological state and impacts in the estuaries. In addition, depending upon the nature of the associations, nitrogen thresholds may be observable. This later proposition gets at the ultimate purpose of such research, that is, to determine how much nitrogen is too much for the types of estuaries studied.

The results can be used to evaluate relative nitrogen loading rates as well as source apportionment for specific estuaries in the context of water quality assessment data. Supplementary material, which contains the nitrogen loading data, is available for each of the estuaries. It should be noted that the results of the model are subject to revision as new data on sources (e.g., atmospheric deposition, fertilizer application rates, and sewage inputs), transport (e.g., transport through soils, septic systems, and leach fields) as well as land use become available; however, the results represent a reliable estimate of first-order nitrogen loading rates.

We therefore used the NLM to first estimate nitrogen loads to the 74 watershed-estuary systems. Second, we compared the NLM based estimate to those of other models. Third, we used the NLM to partition the total load to the 74 estuaries into the relative contributions by the major source categories (direct and indirect atmospheric deposition, wastewater discharge, and fertilizer use) to define their relative contributions in the region and link them to land use.

2. Methods

2.1. Area of study

The watershed-estuary systems are located along the coast of southern New England (USA) and span the shorelines of Connecticut (CT), Rhode Island (RI), and Massachusetts (MA) (Fig. 1). The region's defining anthropogenic characteristic is that it is situated in the major urban corridor from New York City to Boston. Therefore, a significant human population lives and works near the shore. The watersheds are relatively small (mean area = 32 km^2) and the land use types range from 100% natural to 89% residential. The estuaries themselves are small (mean area = 2.3 km^2) and shallow (mean depth = 4.5 m).

2.1.1. Aquatic regime

The study systems are within the Virginian Atlantic marine ecoregion (Wilkinson et al., 2007). This marine ecoregion is characterized by sea-surface temperatures of 2–20 °C (winter) and 15–27 °C (summer), and by currents isolated from the deep waters of the North Atlantic Ocean by the Gulf Stream. This area is further characterized by a wide continental shelf, rocky coastal zones to the north, and salt marshes and sandy beaches to the south.

2.1.2. Terrestrial regime

The southern New England estuarine watersheds lie within the Northeastern Coastal Zone and the Atlantic Coastal Pine Barrens terrestrial ecoregions (EPA, 2006). The Northeast Coastal Zone (NCZ) is characterized by relatively nutrient poor soils and glacially formed lakes. The Atlantic Coast Pine Barrens (APB) is comprised of coarse-grained soils, cool climate, and Northeastern oak-pine. Its climate is milder than the Northeastern Coastal ecoregion to the north (see Supplemental material for data on ecoregions).

In this study we evaluate whether differences in land use/land cover of the Northeast Coastal Zone and the Atlantic Coast Pine Barrens result in geographically distinct nitrogen loading regimes. In addition, because of the degree of land use differs among the three New England States (CT, RI, MA), we also partitioned the data to evaluate whether nitrogen loading from watersheds also differed among these political divisions.

2.2. Components of the model

Nitrogen loading rate values calculated for the study estuaries are based on the application of a published nitrogen loading model (NLM) (Valiela et al., 1997). The NLM provides nitrogen loading rates to watersheds and receiving waters. It considers diffuse, nonpoint source inputs and includes estimates of losses in various compartments of the watershed. The model was developed for Waquoit Bay, Massachusetts, USA, and is considered most applicable to rural-to-suburban watersheds underlain by unconsolidated sandy soils (Valiela et al., 1997).

The original NLM was verified with measured loading and isotope data from several sub-estuaries of Waquoit Bay, MA, USA (Valiela et al., 2000) and further verified with measured loading data from Barnegat Bay, NJ, USA (Bowen et al., 2007a,b). The formulation and use of NLM is available online at http://nload.mbl.edu/. NLM included three major nitrogen inputs to watersheds: 1) Atmospheric deposition to four land use types (natural vegetation, turf, agricultural land, and impervious surfaces); 2) Fertilizer application to two land use types (turf and agricultural land); and 3) Human wastewater nitrogen. In the algorithm the nitrogen is attenuated as it passes through the watershed surface and subsurface zones; this attenuation is dependent upon whether the sources fall on natural vegetation, turf, agricultural, or impervious surfaces within the watershed of each study estuary. The final, or net, input of nitrogen to the estuary is thus the sum of the inputs from sources minus losses in the various land use types in the watershed.

We augmented the NLM with two additional input terms: direct atmospheric deposition to the estuary surface and, in a small number of cases, direct point source inputs from wastewater treatment facilities that discharge directly into the estuary. Any comparisons in this paper with the results from other loading models and literature values were done to compensate for these terms.

It is worth noting that another model, the Estuarine Loading Model (ELM), has been formulated to estimate the mean annual dissolved inorganic nitrogen (DIN) concentration in the estuary itself (Valiela et al., 2004). The ELM takes the output of the original

Fig. 1. Map of locations of study estuaries and their associated watersheds (see Supplemental material for estuarine ID and additional data).

NLM and includes various additional input and loss terms, including direct atmospheric deposition. The ELM produces estimates of DIN, whereas the NLM produces estimates of total dissolved nitrogen (TDN, which include dissolved organic nitrogen); therefore the reader should be careful to make the appropriate transformations to compare NLM (TDN) and ELM (DIN) estimates.

2.2.1. Land use data

Land use data were used for a variety of calculated variables in the application of the NLM. The sources of each of these land use data sets are described in Supplemental material. Ideally, the land use data for all sites, across the three States, would be derived from one data set for comparison purposes; but for the sake of accuracy, the more detailed and current State-developed land use data layers were available only for RI and MA. CT land use data are less specific than those for RI and MA.

In summary, the application of the NLM required delineation of the boundaries of the study estuary, delineation of the watershed for each estuary, acquisition of land use data for each watershed, and aggregation of land use data into the land use types needed for NLM calculations (natural vegetation, recreation, agriculture, commercial, and residential). These steps required use of ArcGIS 9.3® to perform spatial analysis. Results of these examinations were imported into MS Excel® files for additional analyses.

2.2.2. Input terms

Table 1 contains the input categories and parameter magnitudes for the NLM. In addition, the algebraic expressions used to calculate the nitrogen inputs to the watersheds are listed.

2.2.2.1. Atmospheric deposition. Atmospheric deposition of nitrogen to estuaries was divided into direct deposition to the water surface and deposition onto the watershed of the estuary (indirect). Both wet and dry deposition were considered. As noted earlier, the original NLM (Valiela et al., 1997) did not include direct atmospheric deposition; which was included here to more fully estimate the total dissolved nitrogen input to the estuary. Data were used from a recent publication that reported nitrogen levels for both wet and dry deposition in coastal Connecticut (Luo et al., 2002). The sample locations for that study spanned the entire coast of CT along Long Island Sound and were in the same general region of many of the study estuaries. In the Luo et al., study, samples were analyzed for nitrate/nitrite, ammonium, and dissolved organic nitrogen to estimate total air and precipitation derived total nitrogen. To compute atmospheric deposition rates, annual precipitation data for the region (source: http://climvis.ncdc.noaa.gov, average for CT, RI, and MA 1990–2000) were coupled with the published atmospheric nitrogen concentrations. Luo calculated total nitrogen deposition fluxes that ranged from 9 to 23 kg ha^{-1} yr^{-1} which is similar to that published from a meta-analysis by Bowen for the decade of the 1990s of 12.5 kg N ha^{-1} yr^{-1} (Bowen and Valiela, 2001).

2.2.2.2. Fertilizer inputs. Published values were used for fertilizer application rates to lawns, active agriculture, and golf courses as well as lawn areas, number of homes, and fraction of homes that use fertilizer (Table 1 and references therein). Fertilizer application rates were 104, 136, and 115 kg N ha^{-1} yr^{-1}, respectively, for lawns, agriculture lands and recreational (e.g., golf courses) land uses.

Table 1
Model equations and input parameter magnitudes.

Input Category	Included Land Use Types	Nitrogen Load Calculation
Atmospheric Deposition		
Natural Vegetation	Forest, wetlands, natural lands	atmos. dep.[a,b] × Area
Turf	Lawns, golf courses	atmos. dep. × Area
Agricultural Land	Crop land	atmos. dep. × Area
Impervious surfaces[c]	Roofs, driveways	atmos. dep. × Area
Impervious surfaces[d]	Roads, runways, parking lots	atmos. dep. × Area
Fertilizer Application		
Turf	Lawns, golf courses	appl. rate[e] × Area × Fn
Agricultural Land	Crop land	appl. rate × Area
Human Wastewater	Residential land	Human excretion rate × persons per home × # homes

[Rainfall nitrate]	270 μg N L^{-1}
[Rainfall ammonium]	920 μg N L^{-1}
[Rainfall dissolved organic N]	180 μg N L^{-1}
[TDN]	1370 μg N L^{-1}
Ave annual rainfall	48.6 in
Wet to total deposition factor	1.25
Median home size	1915 sq ft
No of stories/home	2
House footprint area	958 sq ft
Average area of roof	1072 sq ft
Average area of driveway	1350 sq ft
Fertilizer N applied to lawns	104 kg N ha^{-1}
Fertilizer N applied to agriculture	136 kg N ha^{-1}
Fertilizer N applied to rec/golf courses	115 kg N ha^{-1}
Average lawn area	0.05 ha
% of homes that use fertilizer	34%
Per capita human N excretion rate	4.8 kg N p^{-1} yr^{-1}
People per house	2.4
# of houses in high density residential areas	8
# of houses in medium-high density residential areas	6
# of houses in medium density residential areas	1.33
# of houses in medium-low density residential areas	0.667
# of houses in low density residential areas	0.5

References/Notes: Valiela et al., 1997 and Luo et al., 2002.
[a] Uses concentration of NO3− NH4+ & DON in local precipitation & yearly rainfall totals to generate atmospheric deposition.
[b] Model includes dry deposition, which is adjustable as a proportion of wet deposition.
[c] Assumes precipitation falling on roofs/driveways subsequently runs off to lawns/natural lands where losses may occur.
[d] Assumes precipitation falling on roads/runways/parking lots is collected in catchment basins & delivered directly to vadose zone.
[e] Uses avg. fertilizer addition rates; F_N refers to fraction of homeowners applying fertilizer.

2.2.2.3. *Wastewater inputs.* Wastewater sources of nitrogen were estimated using information about houses in the watershed, persons per house, and per capita nitrogen excretion rates. The number of houses located on the watershed of each estuary was inferred from the area of residential land use as well as data on the house density per area land use (Table 1). For most of the 74 watershed-estuary systems there was no direct input from wastewater treatment facilities. In those few estuaries with direct point source discharge, nitrogen monitoring data were obtained from the wastewater facilities.

2.2.3. *Loss terms*

Published transport/retention coefficients were applied to each type of land use category (Fig. 2). The nitrogen that comes from the three sources deposited on the watershed was lost, or attenuated, according to processes parameterized by the coefficients in the figure (Valiela et al., 1997). Except for nitrogen deposited on impervious surfaces (100% transported), from 62 to 65% of the indirect atmospherically deposited nitrogen was retained within the watershed (Fig. 2). In contrast, fertilizer nitrogen applied to turf and agricultural land was largely transported to the subsurface (i.e., only 39% lost at the surface).

Human wastewater nitrogen is derived from individual sewage disposal systems (ISDSs) or via sewering through wastewater treatment facilities (WWTFs). Nitrogen from ISDS sources was partially retained in septic tanks and leach fields (40%) as well as septic plumes (34%) through the watershed on its way to the estuary.

WWTFs receive untreated sewage and apply treatment technologies that reduce effluent nitrogen. For those estuaries that have direct discharge, the WWTF nitrogen loading value was calculated from monitoring data from each treatment facility. The computation of total nitrogen from each WWTF was straightforward and therefore the estimate was considered an accurate measure of point source inputs to the affected estuaries. In contrast, the proportion of total population, in a specific watershed, served by the WWTF was not readily available. Due to the lack of sewer-line data at the plat-scale within each watershed, it was not possible to ascertain the relative proportion of point and nonpoint inputs for the eleven (11) affected watersheds. For these estuaries, human waste-derived nitrogen loading was calculated independently, either by assuming that (1) 100% of human waste was watershed-derived (nonpoint source) or by assuming that (2) 100% was derived from WWTFs. The larger of the two values was chosen. This avoided double accounting for human waste-derived nitrogen. Future refinement is needed to evaluate methods of better accounting for the relative proportion of point and nonpoint nitrogen from human waste.

Once nitrogen passes the watershed surface it enters the vadose and aquifer zones. In the vadose zone 61% of the nitrogen was lost and an additional 35% was lost in the aquifer zone (Fig. 2). Evidence suggests that aquifer losses (at least in the upper layers) are likely due to denitrification (cf. Fig. 3 in Bowen et al., 2007a,b).

In summary, overall nitrogen loss/transport varies by the sources of nitrogen through the watershed and into the marine estuary. For example, for every 100 units of nitrogen that came from indirect atmospheric deposition and passed through natural vegetation, only 9 units reached the marine environment, a loss of 91%. In contrast, atmospheric nitrogen that passed through impervious commercial land uses to the subsurface was only reduced by 75%. Additionally, overall, about 74% of the nitrogen derived from human waste was removed between the ISDS and below-ground processes in the watershed.

2.3. *Model assumptions*

The data required to use the NLM to compute nitrogen inputs from the watersheds to the study estuaries is summarized in Table 1. It should be noted that because estimates of loading were based on infrequent land use assessments (from 1992, 1995, and 1999) the estimated loading values were considered representative of the long-term average for the 1990s.

As originally formulated, the NLM algorithm computes nitrogen loads to the water body edge. It does not include any of the other processes that add, remove, and transform nitrogen within estuaries. The reader is directed to the ELM, a complimentary model to the NLM, for consideration of these processes (Valiela et al., 2004). We added direct atmospheric deposition as well as, in a small number of cases, point source inputs that discharge directly into the estuary.

One input not included in the nitrogen loading estimates is from the open end of the estuary derived from the larger ocean system

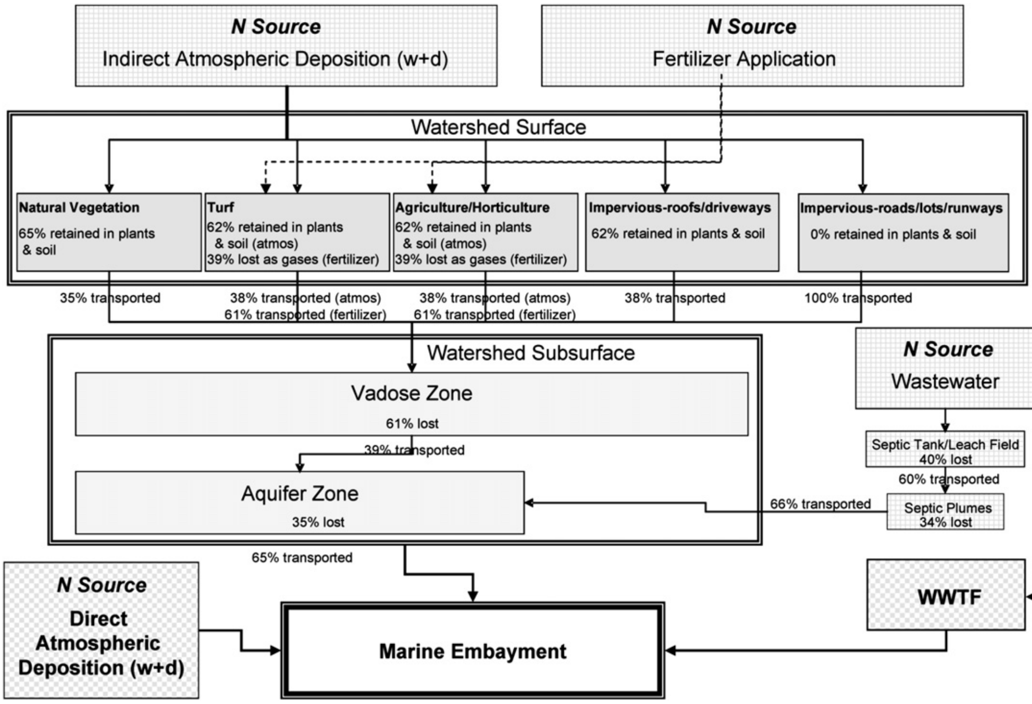

Fig. 2. Schematic of the nitrogen loading model (NLM, with direct atmospheric deposition and point source components added in stippled boxes).

(Nixon et al., 1995). For estuaries that are nested within larger estuaries that have a strong nitrogen signal (e.g., Providence River, Oviatt, 2008), the ecological effects may be ascribed to both local watershed inputs as well inputs from the larger estuarine systems. The NLM is formulated to estimate nitrogen only from the local estuarine watershed to the small estuary; the only nitrogen loading source not from the watershed is atmospheric deposition directly onto the water surface.

Internal nitrogen regeneration from sediments and the water column is not considered in this paper; however, it is taken into account by the ELM. The sediment has been ascribed by others to be a net sink, except during summer periods where it may be a net source (Howes et al., 2003); in either case it is not "new" nitrogen; therefore, it was not included.

In summary, we provide an estimate of loading rates from major anthropogenically-derived watershed sources as well as direct atmospheric deposition. Watershed sources are likely the most appropriate for environmental management of estuarine water quality conditions, simply because these sources are most amenable to watershed-scale remedial action.

Each of the components required to estimate nitrogen loading to the estuaries is subject to uncertainty. Using a bootstrap resampling method for evaluating uncertainty in the components that make up the estimate, Collins et al., 2000, calculated a $\pm$13% uncertainty in the loading estimates derived from the NLM. This assessment of uncertainty should be considered when the NLM is applied in the formulation of subsequent pressure-state models.

The NLM has been verified for embayments of Waquoit Bay, MA, USA (Valiela et al., 2000, 2002). There were two lines of evidence used to verify the model in Waquoit Bay: a comparison with total measured loads and a comparison of predicted wastewater inputs with nitrate δ^{15}N in groundwater samples. The relationship between measured and predicted nitrogen loads described nearly 80% of the variance in the data and was not significantly different from the 1:1 line. Moreover, there was a strong positive relationship between the model predicted fraction of wastewater input and the isotopic composition of groundwater, which converged on the published nitrogen isotopic composition with 100% wastewater inputs. Another study compared NLM + direct atmospheric deposition to Barnegat Bay, NJ, USA with measured nitrogen loading rates (Bowen et al., 2007a,b). The modeled estimate was within 10% of the measured estimates. It is worth noting that Barnegat Bay is in a completely different geographic region from where the NLM was formulated. These separate studies provide compelling evidence of the veracity of the NLM.

3. Results and discussion

3.1. Comparison with other nitrogen estimation approaches and estimates

To provide evidence that supports its application to estuaries beyond where it was verified, the NLM estimates from this study were compared (1) to those of another published nitrogen estimation approach and (2) to those estuaries that have published nitrogen loading rate data.

Nitrogen loading rates calculated using the NLM were compared to loading rates calculated from the USGS New England-SPARROW (NE-SPARROW) model (Moore et al., 2004; Fig. 3A). SPARROW loading estimates were derived by summing the model output for the stream nodes associated with each estuary and adding direct atmospheric deposition and point source inputs, where applicable. Uncertainly analysis of the NLM, applied to the loading estimates (Collins et al., 2000), as well as an uncertainty of $\pm$40% for the nitrogen loading estimates of the NE-SPARROW model (R. Moore USGS, personal communication) are depicted using error bars in Fig. 3.

The estimated nitrogen loading rates for 35 estuaries, common to both the NLM and NE-SPARROW, were relatively similar (Fig. 3A).

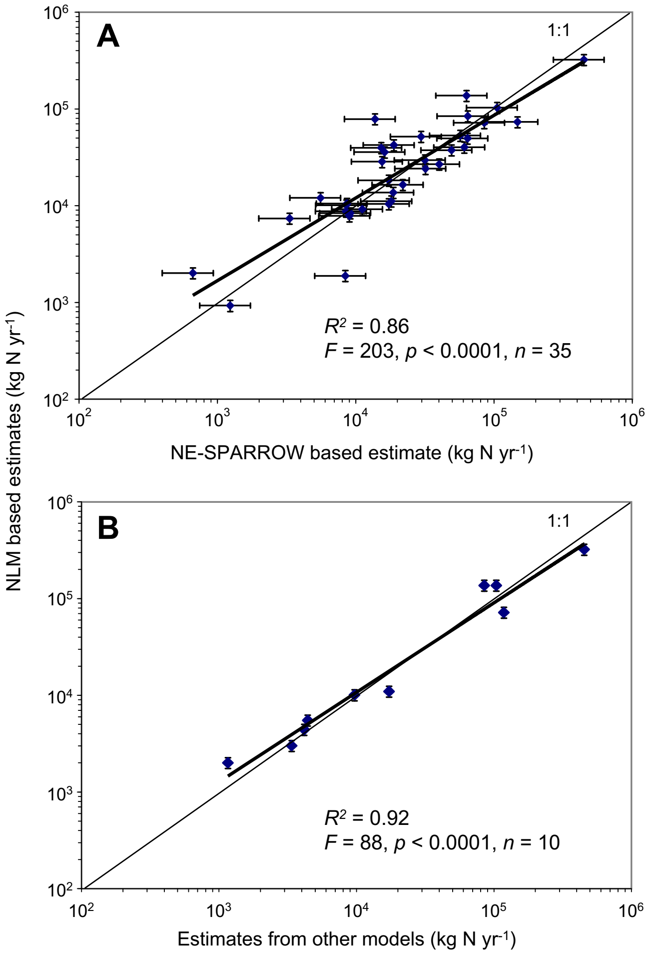

Fig. 3. Regressions showing (A) comparison of NLM and NE-SPARROW nitrogen loading rate estimates (log–log format) and (B) comparison of the estimated loading rates from this study to other published values (log–log format). The thin diagonal line on both figures represents the 1:1 line. Data sources for (B): Pawcatuck River Estuary, CT (Vaudrey, 2008), Greenwich Bay, RI (Granger et al., 2000), Acushnet River, MA (SMAST 2007), Phinneys Harbor, MA (Howes et al., 2006a), Hamblin Pond, MA (Howes et al., 2005), Jehu Pond (Howes et al., 2005), Lagoon Pond, MA (MVC 2000), Sage Lot Pond, MS (Howes et al., 2005), West Falmouth Harbor, MA (Howes et al., 2006b).

The results in this comparison are striking given that the SPARROW approach is very different from the NLM. The SPARROW approach links measured stream transport rates to spatially referenced descriptors of nitrogen sources, land-surface and stream-channel characteristics and the model has been verified by a comparison of measured and observed loading values yielding a relationship that explained 95% of the variance in the data (Moore et al., 2004). In contrast, the NLM doesn't use stream data or explicit spatial referencing but rather integrates all the sources in the watershed. Summarizing, the two approaches yield comparable results and suggest that the NLM provides a reasonably accurate first-order estimate of nitrogen loading rates from small watersheds to their associated estuaries.

A comparison of NLM estimates from this study to other published estimates for ten estuaries in New England shows comparable results (Fig. 3B). Most of the published estimates are based on watershed loading algorithms similar to the NLM. For example, estimates published by Howes et al. (see citations in the figure title) utilize the following process: (a) quantify sources to the land or aquifer, (b) confirm that groundwater transport load has reached the estuary, and (c) quantify nitrogen attenuation that can occur during travel through lakes, ponds, streams and marshes. While strictly not a validation, the relatively close estimates provide additional confidence in NLM estimates.

Based on these analyses as well as the published record, we have confidence that the NLM is an appropriate model for the estimation of total dissolved nitrogen loading rates from small watersheds to estuaries in southern New England. Next we use NLM to estimate total, wastewater, fertilizer, and atmospheric loads to the estuaries in the southern New England region.

3.2. Nitrogen loading rates to southern New England estuaries by source

In addition to its utility in providing total nitrogen loading rates estimates, a useful feature of NLM is that it can identify relative contributions by different sources of nitrogen (e.g., atmosphere, fertilizer, and human waste) (Valiela and Bowen, 2002).

3.2.1. Regional spatial patterns

We looked across the entire southern New England geographic area to determine spatial patterns in the total loading and the relative source proportions of nitrogen to the study estuaries. We utilized frequency distributions to evaluate the spatial trends by terrestrial ecoregion and by State (CT, RI, and MA).

The relative magnitudes of total nitrogen loading to the estuaries differed considerably depending upon how the region was partitioned (Fig. 4, Col. 1). No estuaries in the Atlantic Coast Pine Barrens (APB) had loading rates >500 kg N ha^{-1}; moreover, the largest number of estuaries in this ecoregion had rates less than 50 kg N ha^{-1} (Fig. 4, Col. 1, Row B). This ecoregion contained the estuaries of outer Buzzards Bay, Cape Cod, and the islands which are downstream of the least developed watersheds. The Northeast Coast Zone (NCZ) contained the largest number of estuaries with loading rates in the high range (99–500 kg N ha^{-1} yr^{-1}). One big difference between the two ecoregions is the number of houses on the watersheds. The average house density in the Northeast Coastal Zone was 53% higher than in the Atlantic Coast Pine Barrens. A comparison of the relationship between the number of houses on each watershed (a surrogate for population) and total loads to the estuary reveals that the two ecoregions had the same ratio, but that the NCZ simply had more houses.

Breaking down the distributions by State (Fig. 4, Col. 1, Rows C, D, and E) revealed that generally the estuaries of RI and MA were dominated by loading rates < 500 kg N ha^{-1} yr^{-1}, with only CT having a number of estuaries with annual loadings greater than 500. MA largest categories of estuaries were those with annual loadings of <99 and contained no estuaries in the >500 kg N ha^{-1} category. The slope of the relationship between number of houses on the watershed and loading to the estuaries was greatest for CT followed by MA and then RI, indicating that additional houses on CT watersheds will have a greater nitrogen loading impact than additional houses on the watersheds of the other two States.

Across the study region, the majority of estuaries had less than 50% of their total nitrogen inputs from direct atmospheric deposition (Fig. 4, Col. 2). Moreover, all of the estuaries in CT fell into this category (Row C). However, more estuaries in the APB, and specifically in RI and MA, had a larger proportion of nitrogen input from this source (Fig. 4, Col. 2, Rows B, D, and E). The magnitude of direct atmospheric deposition was a function of regional deposition rates as well as estuary areas.

Indirect atmospheric deposition, which was a function of watershed size and loss processes as well as atmospheric deposition rates, generally comprised <25% of the total nitrogen inputs to the study estuaries (Fig. 4, Col. 3, all rows). Slightly greater numbers of estuaries in CT were in the 25–50% and 51–75% proportion categories (Fig. 4,

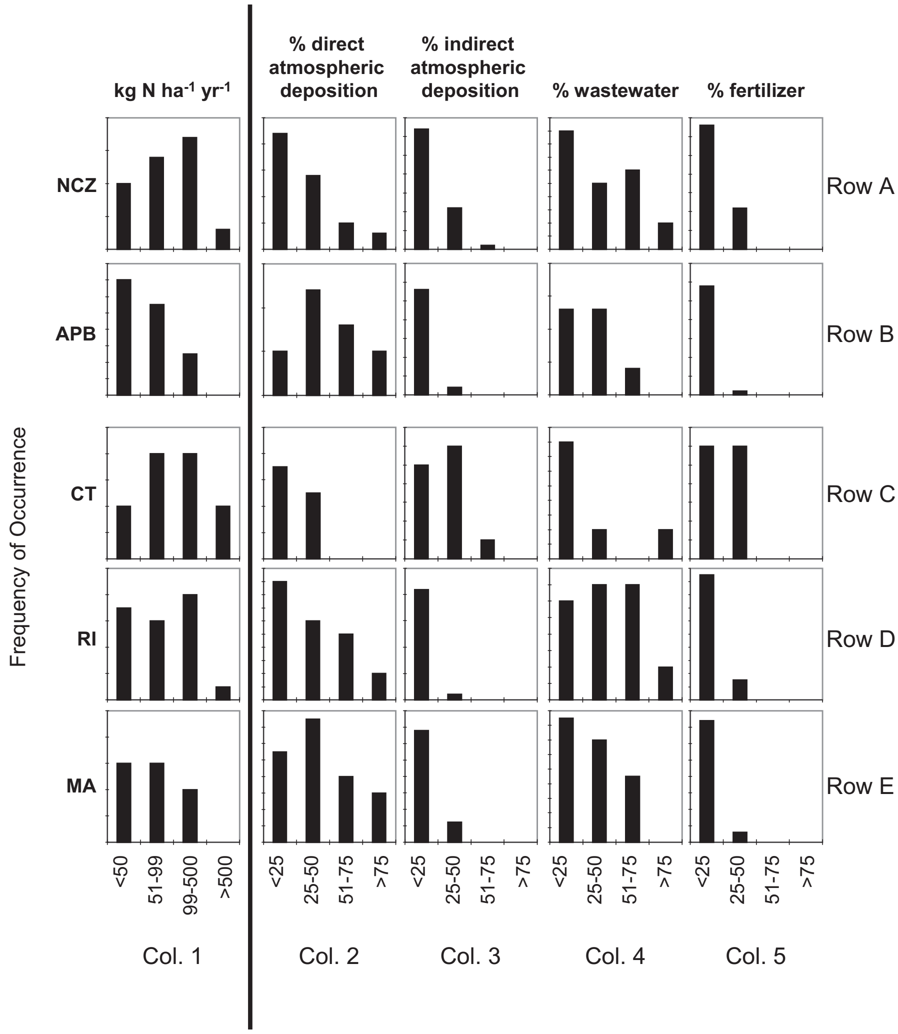

Fig. 4. Graphs showing frequency distributions of the total nitrogen loading rates (Col. A) of the watershed-estuary systems for the southern New England region grouped by terrestrial ecoregion: Northeast Coastal Zone (NCZ) and Atlantic Coast Pine Barrens (APB), (Rows A and B, respectively), or by State (Rows C, D, and E). Col. 2–5 show the frequency distributions of the percent of the watershed-estuary systems in each of the source categories (i.e., direct atmospheric deposition, indirect atmospheric deposition, wastewater and fertilizer, respectively) grouped similarly.

Col. 3, Row C). Natural vegetation in the watersheds (i.e., forests) receive both wet and dry atmospheric nitrogen deposition which will then be attenuated at the surface and subsurface of the watershed by fixation, burial, and denitrification and only a small fraction will enter the estuary. Not surprising, the relationship between the area of forests and the total nitrogen loading to the estuaries was essentially the same between the two ecoregions and for MA and CT. RI, however, had a stronger relationship, likely because its estuaries had the smallest watershed areas so actual attenuation might be less that what was applied in the NLM.

One can conclude that the region exhibited spatial heterogeneity in the relative magnitude of wastewater derived nitrogen (Fig. 4, Col. 4). There was a clear difference in the dominance of wastewater between the two ecoregions (Fig. 4, Col. 4, Rows A and B). The magnitude of wastewater nitrogen inputs to an estuary in NLM was a function of population density (i.e., house density), as well as watershed loss processes. It is no surprise, given the house densities, that NCZ had a greater number of estuaries with >50% wastewater inputs (Fig. 4, Col. 4, Row A). Looking more closely, it was apparent that RI (which are mostly within the NCZ) had a large number of estuaries dominated by wastewater inputs (i.e., >50%, Fig. 4, Col. 4, Row D). This is consistent with the fact that RI had the highest house density of all three States (mean = 263 houses km^{-2} watershed area).

The magnitude of fertilizer derived nitrogen inputs was a function of the area of agriculture land uses, fertilizer application rates, as well as watershed loss processes. Throughout the entire study region, agriculture land use area was a relatively minor component of the total watershed, with the majority of estuaries exhibiting less than 25% of the total nitrogen from this source (Fig. 4, Col. 5). CT was the one exception where there were a larger number of estuaries that had between 25 and 50% of their total nitrogen from fertilizer inputs (Fig. 4, Col. 5, Row C).

In summary, there were regional differences in total inputs as well as the relative source terms over the study region. These differences were a function of the human activities on the watersheds (land use) as well as the processes that attenuate nitrogen en route to the estuary. It appears that estuaries in CT were more susceptible to additional watershed development than estuaries in MA and RI; although all estuaries in the region are affected by wastewater from watershed development. This information has management implications for source reduction (Bowen and Valiela, 2004) as well as best management practices in the watersheds for the attenuation of nitrogen.

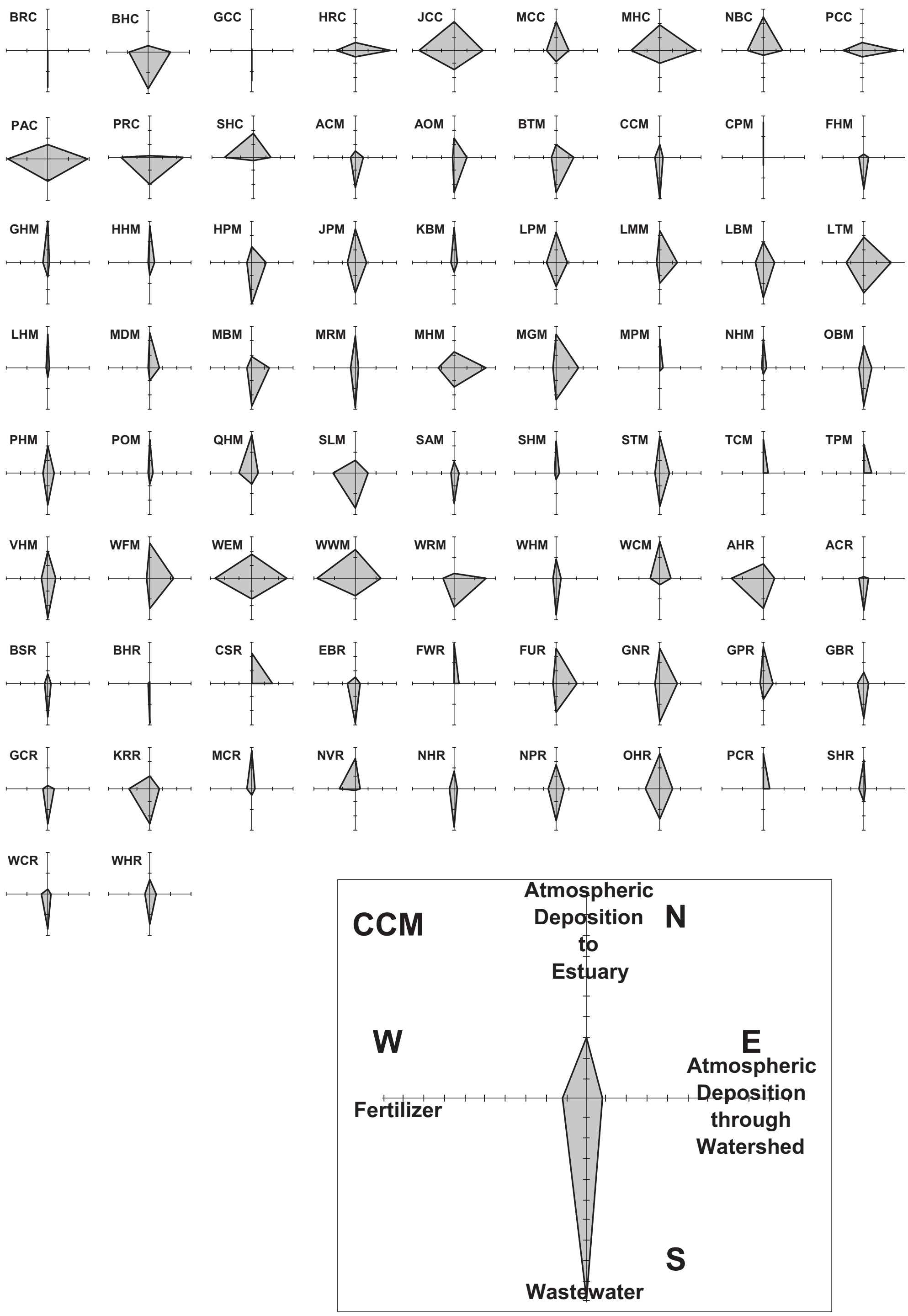

Fig. 5. Spider diagrams of the relative importance of the four nitrogen sources to the individual watershed-estuary systems (all values are normalized to the maximum value for ease of source comparison). Atmospheric deposition to estuary, atmospheric deposition through watershed, wastewater, and fertilizer are located on the N, E, S, and W compass points of the graph, respectively. The three capital letters in the upper left of each figure denote the estuarine ID (see Supplemental material).

3.2.2. *Relative source magnitudes*

Knowledge of the relative magnitude of nitrogen sources is useful to evaluate potential management actions that may be required to meet nitrogen reduction goals for each watershed-estuary system. Spider plots depict the relative importance of the four components of nitrogen inputs to each of the estuaries (Fig. 5).

Those watershed-estuary systems that have dominance in the “N” and “E” points of the spider plots are not amenable to local nitrogen source reductions because these systems are dominated by direct or indirect atmospheric deposition. Since atmospheric nitrogen source control is considered a regional scale issue, local communities cannot significantly reduce this source. For example, plans for lowering nitrogen inputs to Long Island Sound included an 18% reduction in atmospheric nitrogen deposition through the application of national air-quality standards in their reduction scenarios, because no local controls are feasible owing to the large airshed of the atmospheric contribution (NYSDEC and CTDEP, 2000).

Those systems that have a large signal in the "S" and "W" points of the spider plots are more amenable to local reductions since these are dominated by human wastewater and fertilizer nitrogen inputs. In these cases, limits on point source effluent as well as nonpoint source management practices within the watersheds would seem to best reduce nitrogen inputs to estuaries.

The results of these analyses were combined to obtain an aggregate assessment of the four sources of nitrogen to the watershed-estuary systems. The overall order of importance for the entire region covered by the 74 watershed-estuary systems was: wastewater ≈ direct atmospheric deposition > indirect atmospheric deposition > fertilizer inputs (Fig. 6).

The combination of direct and indirect (wet + dry) atmospheric deposition is, on average, the largest input of nitrogen to the study estuaries. The range in indirect (<1–52%, mean = 16%) and direct (1–89%, mean = 37%) atmospheric deposition was similar to other published values, although exact comparisons are not always feasible. Paerl et al. (2001), reported a range in direct wet + dry atmospheric deposition of <5–70% for estuaries around the US and Europe; indirect inputs were not reported. The range in direct atmospheric deposition was a function of the regional deposition rate and the area of the estuary itself. While indirect atmospheric deposition is mediated by watershed processes such as fixation, denitrification, and other loss terms, direct deposition has no such mediation and as such can stimulate algal blooms directly. Therefore, regional-scale emission reductions of both stationery and fixed atmospheric emission sources would reduce their effects on estuaries.

Finally, wastewater inputs spanned from zero to nearly 98% of the total nitrogen (mean = 36%). This source clearly has significant local management implications which may require reductions of nitrogen from ISDS. Fertilizer inputs, in contrast, were generally low (mean = 12%) and thus agricultural mitigation activities are not expected to be a major management priority for this region.

In summary, the ability to evaluate the relative importance of nitrogen sources illustrates that the NLM can be used to prioritize watershed-based management scenarios as well as provide scientific justification for regional reductions in atmospheric emissions.

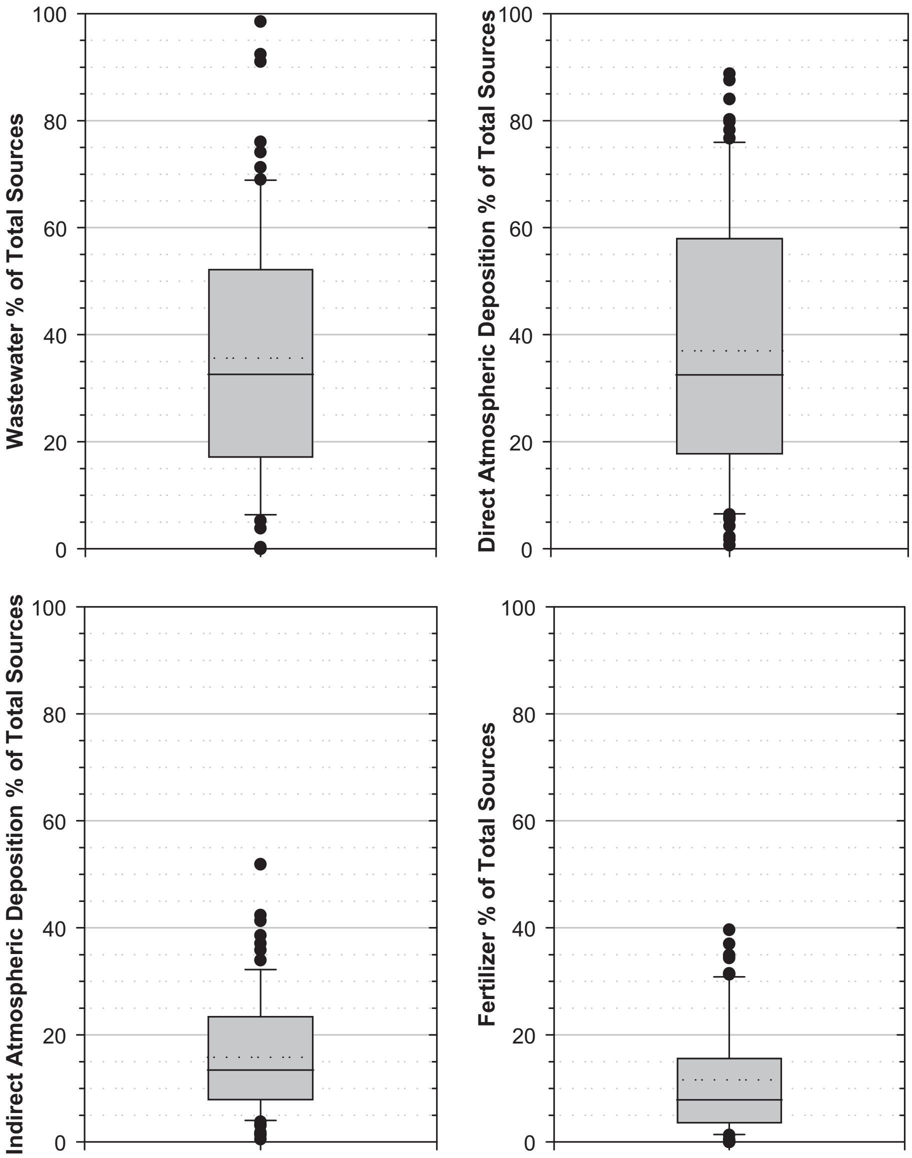

Fig. 6. Box-and-whisker plots showing the aggregated statistical distribution of the four nitrogen sources for the entire region's study estuaries. These figures show the lowest and highest values, the 10th, 25th 50th, 75th, as well as the mean (dotted line) for each of the sources for the entire set of study estuaries.

3.3. Total nitrogen inputs to southern New England estuaries

The total loading to an estuary will be directly proportional to watershed sources and inversely proportional to watershed retention processes. The size of the estuary itself will mediate the nitrogen load. All other factors being equal, the same load going into a large estuary will have less of an effect than the same amount going into a small estuary. When comparing loading between estuaries one needs to consider these scaling factors. In this study the maximum nitrogen loading rate to the study estuaries was estimated to be 370,000 kg N yr^{-1}(Table 2); those reported by others ranged from 980,000 to 148 × 10^6 kg (Castro et al., 2003; Whitall et al., 2007). Therefore, when none of these scaling factors are considered, estimated loading rates to the southern New England study estuaries were all significantly lower than those reported in the literature, leading to the erroneous expectation that there will be only limited symptoms of eutrophication in the study estuaries.

When estuarine areal differences are included, (by calculating loading on an areal basis, Table 2) the nitrogen loading rates were more similar (mean = 167 kg N ha^{-1} estuarine area yr^{-1}), yet still lower than, other US estuaries (mean = 893). However, to compensate for scale differences in watershed size, nitrogen yields were calculated, i.e., nitrogen export from the watersheds (after attenuation) normalized to watershed area (kg N ha^{-1} watershed area yr^{-1}). When evaluated in this manner, the nitrogen yields for the southern New England estuaries were similar to other US estuarine watersheds (Table 2). For example, the loading rate to the Chesapeake Bay has been reported to be 13.5 kg N ha^{-1} watershed area yr^{-1}; this is slightly lower than the value for southern New England estuaries (mean = 17). In fact, when evaluated in this manner, the average southern New England estuary had higher nitrogen yields than Long Island Sound (12.9), Great Bay NH (6.8) and Charleston Harbor SC (13.5), all of which exhibit symptoms of eutrophication.

Considering the large variation in nitrogen loading for the 74 small watershed-estuary systems in this study (RSD from 140 to 250%, Table 2) it seems likely that upscaling loads to larger spatial units might best be done by adding loads from separate watersheds within regions rather than by using larger regional and continental scale extrapolations such as Global NEWS (Mayorga et al., 2010). The need to obtain separate estimates from component watersheds is a burdensome requirement, so that examining the statistical constraints on the corresponding upscaling approaches might be a useful next step to develop the utility of the present study.

The results of the application of the NLM to the 74 watershed-estuary systems provide an understanding of the magnitude of nitrogen loading to estuaries in southern New England, but alone are insufficient to determine how much nitrogen is too much. What is lacking is the associated expression of the effects along the gradient of nitrogen inputs. According to common understanding of how nutrients affect estuaries, at levels below some critical loading, nutrients provide benefits to the healthy structure and function of estuaries. Estuaries are dynamic environments that can assimilate nutrients depending upon their geomorphic and hydrodynamic properties which affect the ability to dilute and flush nutrient loads. Knowledge of estuarine susceptibility to nutrients and the associated expressions of effects is important (NRC, 2000). The NLM provides one essential component in the development of quantitative empirical pressure-state relationships suitable to determine how much nitrogen is too much. The other essential components are data on effects or symptoms of eutrophication, such as, for example, water clarity, chlorophyll-a magnitude as well as indicators tied directly to designated uses, such as extent of hypoxia and extent of ecologically important resources such as seagrasses. On a national basis one needs to place the pressure-state models into a classification schema that allows the grouping of US estuaries according to important geomorphic and hydrodynamic properties, so that class-specific pressure-state models may be developed. In addition, because of the close coupling between watershed and estuarine condition (Paul et al., 2002), watershed characteristics including slope, land use, pollution sinks and size will factor into pollution gradient assessment and experimental design (Fu et al., 2005; Rodriguez et al., 2007).

The NLM provides a useful tool to evaluate watershed-scale management practices. However, before watershed management scenarios are ultimately explored, a water quality goal, for example a critical load, quantifying how much nitrogen is too much, is required.

Table 2
Summary of nitrogen loading rates for New England and other US estuaries.

	kg N yr^{-1}	kg N ha^{-1} estuarine area yr^{-1}	kg N ha^{-1} watershed area yr^{-1}
Source		This study	
Minimum	43	24	3.1
10th percentile	592	28	5.6
25th percentile	3030	36	7.0
50th percentile	10,500	65	12
75th percentile	29,600	117	19
90th percentile	67,400	311	31
Maximum	365,000	3310	155[a]
Arithmetic mean	29,400 ± 59,100 (24,800)[a]	167 ± 415 (124)[a]	19 ± 26 (17)[a]
Count	74	74	74
Source	(Whitall et al., 2007)[b]	(Whitall et al., 2007)[c]	(Castro et al., 2003)
Casco Bay	983,506	23	5.3
Great Bay	1,663,490	354	6.8
Merrimack River	10,279,096	6424	9.5
Massachusetts Bay	15,476,565	202	49.0
Buzzards Bay	1,066,945	17	21.8
Narragansett Bay	8,444,631	203	27.2
Long Island Sound	39,856,585	122	12.9
Hudson R/Raritan Bay	76,222,208	954	24.0
Barnegat Bay			7.3
Delaware Bay	51,394,927	248	20.2
Chesapeake Bay	147,839,494	270	13.5
Pamlico Sound	45,372,756	1004	18.2
Wynah Bay			12.7
Charleston Harbor			13.5
St Helena Sound			5.7
St Catherines–Sapelo			2.3
Altamaha Sound			9.4
Indian River			29.1
Charlotte Harbor			18.1
Tampa Bay			26.9
Apalachee Bay			5.6
Apalachicola Bay			10.0
Mobile Bay			8.5
West Mis. Sound			9.1
Barataria Bay			8.3
Terrebonne–Timbalier Bays			10.6
Calcasieu River			11.7
Sabine River			9.3
Galveston Bay			16.5
Matagorda Bay			4.0
Corpus Christi Bay			2.4
Upper Laguna Madre			1.0
Lower Laguna Madre			8.7
Arithmetic Mean ± SD	36,200,000 ± 44,600,000	893 ± 1870	13.2 ± 9.69

[a] Black Rock Harbor (BRC) exceeded the highest reported US value. This was due to the large WWTF discharge and a small watershed (means with BRC excluded are in parentheses).
[b] Calculated.
[c] Corrected from references.

4. Conclusions

Watershed loading models such as the NLM are useful to estimate nitrogen loading rates to multiple estuaries in comparative studies. Due to a favorable assessment between the NLM and the NE-SPARROW model, published calibration results, and comparable published values for study estuaries, the NLM is capable of providing a first-order, loading estimate for small-to-medium sized estuaries in southern New England.

Results of this study indicate that estuaries of southern New England may be exposed to nitrogen loading rates from <50 to 370,000 kg yr^{-1} (<30 to 3300 kg ha^{-1} yr^{-1}) with watershed yields that range from <5 to 160 kg ha^{-1} yr^{-1}. The dominant sources of nitrogen are: wastewater ≈ direct atmospheric deposition > indirect atmospheric deposition > fertilizer. Across the region, the combined inputs of direct and indirect atmospheric deposition rival that of wastewater inputs. However, results varied dramatically for each individual estuary, where other sources, such as wastewater inputs can dominate.

NLM results show that the region's estuaries have a large range in nitrogen inputs, and as such are suitable for field-based studies requiring a nitrogen gradient. Moreover, because of the large number of study estuaries for which nitrogen loading has been estimated, it is possible to select more than one watershed-estuary system with similar nitrogen loading so that field-replication (at least for loading) can be obtained. Obviously, other characteristics are needed in addition to a large nitrogen gradient, including comparability with, or gradients in, hypsography, substrate types, and other physical-ecological variables of the estuary.

This paper illustrates the application of the previously published and verified NLM to estimate and evaluate total dissolved nitrogen loading rates as well as the relative source strengths for a large number of estuaries in southern New England.

Acknowledgments

We thank the following people for providing data and insights in support of this work: Paul Stacey and Ann Straut-Esden (CT Department of Environmental Protection), John Mullaney (US Geological Survey), Scott Nixon (University of Rhode Island), Scott Duerr (Westerly, RI Wastewater Treatment Facility), Russell Isaac, and Steve Halterman (MA Department of Environmental Protection). We also thank the peer reviewers for their contributions in making the paper more concise and scientifically focused.

The research described in this article was funded by the US Environmental Protection Agency, but it has not been subjected to Agency review. It does not necessarily reflect the views of the Agency. Mention of trade names or commercial products does not constitute endorsement or recommendation for use. AED contribution number AED-09-036.

Appendix. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecss.2010.06.006.

References

Bowen, J.L., Kroeger, K.D., Tomasky, G., Pabich, W.J., Cole, M.L., Carmichael, R.H., Valiela, I., 2007a. A review of land–sea coupling by groundwater discharge of nitrogen to New England estuaries: mechanisms and effects. Applied Geochemistry 22, 175–191.

Bowen, J.L., Ramstack, J.M., Mazzilli, S., Valiela, I., 2007b. NLOAD: an interactive, web-based modeling tool for nitrogen management in estuaries. Ecological Applications 17, S17–S30.

Bowen, J.L., Valiela, I., 2004. Nitrogen loads to estuaries: using loading models to assess the effectiveness of management options to restore estuarine water quality. Estuaries 27, 482–500.

Bowen, J.L., Valiela, I., 2001. Historical changes in atmospheric nitrogen deposition to Cape Cod, Massachusetts, USA. Atmospheric Environment 35, 1039–1051.

Castro, M.S., Driscoll, C.T., Jordan, T.E., Reay, W.G., Boynton, W.R., 2003. Source of nitrogen to estuaries in the United States. Estuaries 26, 803–814.

CBP, 2000. 2000 Chesapeake Bay Agreement. Chesapeake Bay Program Office, Annapolis, MD, 13 pp.

Cloern, J.E., 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223–253.

Collins, G., Kremer, J.N., Valiela, I., 2000. Assessing uncertainty in estimates of nitrogen loading to estuaries for research, planning, and risk assessment. Environmental Management 25, 635–645.

Epa, 2006. Level III ecoregions of the conterminous United States. http://www.epa.gov/wed/pages/ecoregions/level_iii.htm (1 of 3) 11/3/2006 5:21:32 PM.

Farley, K.J., Rangarajan, S., 2006. A Decision Support Framework to Facilitate Nitrogen Load Reductions in the Long Island Sound (LIS) Watershed. US Environmental Protection Agency, Long Island Sound Office, Stamford, CT, LI 97286104-0, 111 pp.

Frink, C.R., 1991. Estimating nutrient exports to estuaries. Journal of Environmental Quality 20, 717–724.

Fu, B.-J., Zhao, W.-W., Chen, L.-D., Liu, Z.-F., Lü, Y.-H., 2005. Eco-hydrological effects of landscape pattern change. Landscape and Ecological Engineering 1, 25–32.

Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P., Sutton, M.A., 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. Science 320, 889–892.

Georgas, N., Rangarajan, S., Farley, K.J., Jagupilla, S.C.K., 2009. AVGWLF-based estimation of nonpoint source nitrogen loads generated within Long Island Sound subwatersheds. Journal of the American Water Resources Association 45, 715–733.

Granger, S., Brush, M., Buckley, B., Traber, M., Richardson, M., Nixon, S.W., 2000. An assessment of eutrophication in GreenwichBay. Paper No. 1. In: Schwartz, M. (Ed.), Restoring Water Quality in GreenwichBay: a Whitepaper Series. Rhode IslandSea Grant, Narragansett, vol. 1, p. 20.

Greening, H., Janicki, A., 2006. Toward reversal of eutrophic conditions in a subtropical estuary: water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. Environmental Management 38, 163–178.

Howes, B., Kelley, S., Ramsey, J., Samimy, R., Schlezinger, D., Eichner, E., 2006a. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Phinneys Harbor – Eel Pond – Back River System, Bourne, Massachusetts. SMAST/DEP Massachusetts Estuaries Project. Massachusetts Department of Environmental Protection, Boston, MA, 142 pp.

Howes, B., Samimy, R., Schlezinger, D., Kelley, S., Ramsey, J., Eichner, E., 2003. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for Stage Harbor, Sulphur Springs, Taylors Pond, Bassing Harbor, and Muddy Creek, Chatham, Massachusetts. SMAST/DEP Massachusetts Estuaries Project. Massachusetts Department of Environmental Protection, Boston, MA, 250 pp.

Howes, B., Samimy, R., Schlezinger, D., Kelley, S., Ramsey, J., Eichner, E., 2006b. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for West Falmouth Harbor, Falmouth, Massachusetts. SMAST/DEP Massachusetts Estuaries Project. Massachusetts Department of Environmental Protection, Boston, MA, 161 pp.

Howes, B., Samimy, R., Schlezinger, D., Kelley, S., Ramsey, J., Ruthven, T., Eichner, E., 2005. Linked Watershed-Embayment Model to Determine Critical Nitrogen Loading Thresholds for the Quashnet River, Hamblin Pond, and Jehu Pond, in the Waquoit Bay System of the Towns of Mashpee and Falmouth, MA. SMAST/DEP Massachusetts Estuaries Project. Massachusetts Department of Environmental Protection, Boston, MA, 149 pp.

Luo, Y., Yang, X., Carley, R.J., Perkins, C., 2002. Atmospheric deposition of nitrogen along the Connecticut coastline of Long Island Sound: a decade of measurements. Atmospheric Environment 36, 4517–4528.

Mayorga, E., Seitzinger, S.P., Harrison, J.A., Dumont, E., Beusen, A.H.W., Bouwmand, A.F., Fekete, B.M., Kroeze, C., Drecht, G.V., 2010. Global Nutrient Export from WaterSheds 2 (NEWS 2): model development and implementation. Environmental Modeling & Software 25, 837–853.

Moore, R.B., Johnston, C.M., Robinson, K.W., Deacon, J.R., 2004. Estimation of Total Nitrogen and Phosphorus in New England Streams Using Spatially Referenced Regression Models. United States Geological Survey, Scientific Investigations Report 2004–5012, 42 pp.

MVC, 2000. Nutrient Loading to Lagoon Pond. Martha's Vineyard Commission, 109 pp.

Nixon, S.W., 1995. Coastal marine eutrophication: a definition, social causes, and future concerns. Ophelia 41, 199–219.

Nixon, S.W., Granger, S.L., Nowicki, B.L., 1995. An assessment of the annual mass balance of carbon, nitrogen, and phosphorus in Narragansett Bay. Biogeochemistry 31, 15–61.

NRC, 2000. Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution. National Academy Press, National Academy of Sciences, Committee on the Causes and Management of Eutrophication, National Research Council (NRC), Washington, DC.

NYSDEC, CTDEP, 2000. A Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound. Connecticut Department of Environmental Protection, Hartford, 49 pp.

Oviatt, C.A., 2008. Impacts of nutrients on Narragansett Bay productivity: a gradient approach. In: Desbonnet, A., Costa-Pierce, B.A. (Eds.), Science for Ecosystem-based Management Narragansett Bay in the 21st Century. Springer New York, New York, pp. 523–543.

Paerl, H.W., Boynton, W.R., Dennis, R.L., Driscoll, C.T., Greening, H.S., Kremer, J.N., Rabalais, N.N., Seitzinger, S.P., 2001. Atmospheric deposition of nitrogen in coastal waters: biogeochemical and ecological implications. In: Valigura, R.A., Alexander, R.B., Castro, M.S., Meyers, T.P., Paerl, H.W., Stacey, P.E., Turner, R.E. (Eds.), Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective. American Geophysical Union, Washington, D.C, pp. 11–52.

Paul, J.F., Comeleo, R.L., Copeland, J., 2002. Landscape metrics and estuarine sediment contamination in the Mid-Atlantic and southern New England regions. Journal of Environmental Quality 31, 836–845.

Reckhow, K.H., Beaulac, M.N., Simpson, J.T., 1980. Modeling Phosphorus Loading and LakeResponse under Uncertainty: a Manual and Compilation of Export Coefficients. US Environmental Protection Agency, Washington, DC, 224 pp.

Rodriguez, W., August, P., Wang, Y., Paul, J., Gold, A., Rubinstein, N., 2007. Empirical relationships between land use/cover and estuarine condition in the Northeastern United States. Landscape Ecology 22, 403–417.

SMAST, 2007. Preliminary Hydrodynamic and Water Quality Analysis of New Bedford Inner Harbor/Acushnet River, New Bedford, Massachusetts (draft). Louis Berger Group, I., University of Massachusetts Dartmouth School of Marine Science and Technology, Applied Coastal Research and Engineering, Inc., Dartmouth, MA, 136 pp.

Valiela, I., Bowen, J.L., 2002. Nitrogen sources to watersheds and estuaries: role of land cover mosaics and losses within watersheds. Environmental Pollution 118, 239–248.

Valiela, I., Bowen, J.L., Kroeger, K.D., 2002. Assessment of models for estimation of land-derived nitrogen loads to shallow estuaries. Applied Geochemistry 17, 935–953.

Valiela, I., Collins, G., Kremer, J., Lajtha, K., Geist, M., Seely, B., Brawley, J., Sham, C.H., 1997. Nitrogen loading from coastal watersheds to receiving estuaries: new method and application. Ecological Applications 7, 358–380.

Valiela, I., Geist, M., McCelland, J., Tomasky, G., 2000. Nitrogen loading from watersheds to estuaries: verification of the WaquoitBay nitrogen loading model. Biogeochemistry 49, 277–293.

Valiela, I., Mazzilli, S., Bowen, J., Kroeger, K., Cole, M., Tomasky, G., Isaji, T., 2004. ELM, an estuarine nitrogen loading model: formulation and verification of predicted concentrations of dissolved inorganic nitrogen. Water Air and Soil Pollution 157, 365–391.

Vaudrey, J.M.P., 2008. Establishing Restoration Objectives for Eelgrass in Long Island Sound Part II: Case Studies. University of Connecticut, Department of Marine Sciences, Groton, CT, 59 pp.

Whitall, D., Bricker, S., Ferreira, J., Nobre, A.M., Simas, T., Silva, M., 2007. Assessment of eutrophication in estuaries: pressure–state–response and nitrogen source apportionment. Environmental Management 40, 678–690.

Wilkinson, T., Bezaury-Creel, J., Hourigan, T., Wiken, E., Madden, C., Padilla, M., Agardy, T., Janishevski, L., Morgan, L., 2007. Marine Ecoregions of North America. The Commission for Environmental Cooperation, Montreal, Canada, 195 pp.